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Ecology and Assessment of Warmwater Streams: Workshop Synopsis

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Ecology and Assessment of Warmwater Streams: Workshop Synopsis

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Contents

Preface

Natural resource agencies are frequently confronted by applications for permits and licenses for developments and actions that affect the aquatic resources of warmwater streams and rivers. At this time, there are no generally accepted methods for assessing impacts and developing mitigation measures for warmwater streams. To begin correcting these deficiencies, U.S. Fish and Wildlife Service biologists and their State agency colleagues need state-of-the-art knowledge on (1) warmwater stream ecology, emphasizing fish and habitat; and (2) existing methods for assessing stream quality and impact.

The need for current information on the ecology and assessment of warmwater streams emerged from the U.S. Fish and Wildlife Service field offices in the Southeast and their State agency colleagues. In response to this need, Warren T. Olds (Assistant Regional Director for Fish and Wildlife Enhancement) and William K. Seitz (Office of Information Transfer) decided in late 1987 that a workshop would be held for biologists involved in managing stream resources and assessing impacts of developments on streams and rivers. The scope and format of the workshop were determined in January 1988 by a steering committee composed of the following persons: Mark B. Bain, Alabama Cooperative Fish and Wildlife Research Unit; John W. Baumeister and James D. Brown, U.S. Fish and Wildlife Service Region 4 Office; Reis Collier, U.S. Fish and Wildlife Service Field Office—Raleigh, North Carolina; Larry E. Goldman, U.S. Fish and Wildlife Service Field Office—Daphne, Alabama; Sam Hamilton, U.S. Fish and Wildlife Service Office of Federal Activities; James Kirkwood, U.S. Fish and Wildlife Service Region 4 Office; and James Layzer, Tennessee Cooperative Fishery Research Unit.

The workshop was held at Lakepoint State Park Resort, Eufaula, Alabama, on 26–27 October 1988. The agenda of the workshop was composed of presentations by two groups of biologists: stream ecologists with extensive basic research experience, and applied research biologists experienced in assessing effects on warmwater streams. There was no intent to merge the basic and applied research; rather the aim was to contrast ecological knowledge with existing impact assessment methods. Biologists and ecologists were invited from Service offices in the Southeast, State natural resource agencies in 11 southeastern States, and Service research and regional offices nationwide. Workshop attendees were about equally divided among these three groups.

This synopsis, which follows the format of the workshop, provides an overview of the topics covered with a list of detailed information sources and expert contacts. Speakers summarized their main points and assembled a list of key papers and books that complement the material they covered or that contain information used in their presentation. Together, the summaries and information sources provide a beginning point for biologists who need current and detailed information on warmwater stream ecology and assessment methodologies.

Acknowledgments

The Office of Information Transfer provided funding for the workshop and this synopsis. D. Parrotte and his staff at the Auburn University Continuing Education Office handled workshop arrangements and administration. The Aquatic Systems Branch of the National Ecology Research Center typed and assembled the first draft of this manuscript. Many subsequent versions were corrected and revised by J. Christian of the Alabama Cooperative Fish and Wildlife Research Unit. P. Angermeier, H. Bart, J. Boltz, C. Couret, and R. Krska reviewed the manuscript.

The workshop sponsors and I thank all speakers for contributing their time and effort to prepare oral presentations and written summaries. I especially appreciate the Fish and Wildlife Service speakers, all of whom chose to contribute to this effort using funding and time from their own programs. This cooperative attitude reflects an unusual degree of commitment by Service people.

Overview

Ecological Research and Impact Assessment: Complementary but Different Endeavors

by

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While both science and impact assessment rely on technical studies, they differ in fundamental goals, approach and scope, and final products. Research and management biologists come from similar educational backgrounds, share similar career interests, and have a high regard for biological knowledge. However, varied philosophy and interests emerge due to different work settings and responsibilities. Science is defined as systematized knowledge derived from observation, study, and experimentation carried on in order to determine the nature or principles of what is being studied. The basic approach to study is the scientific method: observation, hypothesis formation, hypothesis testing, deriving results, and interpreting findings relative to principle or theories. Impact assessment is the process of documenting the important consequences of proposed actions by (1) objective analyses of current and predicted conditions and (2) subjective evaluation of the significance of predicted changes. In contrast to the scientific method, the impact assessment method uses distinctly different steps: reviewing proposed actions, documenting baseline conditions, identifying possible impacts, predicting changes, documenting significant impacts, and formulating recommendations. The definitions and methods of both science and impact assessment reflect the fundamental differences between these endeavors.

Other aspects of science and assessment also distinguish the professions. Most notable among these is the interpretation of the term *significance*. To scientists, the term refers to the probability level associated with rejecting a true null hypothesis; in practice, a result is significant if the data deviate enough (probability of occurrence ≤ 0.05) from the prediction of the null hypothesis to be considered different. To assessment biologists, significance is determined by a subjective determination that in practice depends largely on agency policies, public concerns, legal standards, personal preferences, and past case histories. The scope of study differs between the professions. Scientists work hard to narrow the number of factors affecting their observations, whereas assessment studies attempt to assemble a diverse array of information, data, and past agency actions to defend a recommendation. Science progresses through the peer-reviewed literature, advancement and replacement of theories and principles, and university education programs. Impact assessment lacks modes of progress due to the limited nature of professional impact assessment journals, dependence on changing agency policies and laws, lack of formal educational programs, and low emphasis on disseminating assessment study findings.

Although science and assessment differ in many ways, the professions complement one another and benefit from exchange of needs and information. Natural resource agencies need reliable and accepted information, methods, and principles from scientists. Most important are the key factors, processes, mechanisms, and structural properties that represent the essential characteristics and functions of species and biological systems. In addition, when this type of information is reduced to accepted basics, science is directly serving natural resource agency biologists. To enhance the value of science to natural resource agencies, assessment biologists should convey information on the type of scientific studies that are most needed: intensive studies of a few factors or extensive studies of complex system patterns, key biological levels of study (species to ecosystems), geographic scale, habitat types, and duration (short- or long-term). Each profession benefits by understanding the difference and needs of the other so that information can be exchanged effectively for mutual benefit.

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Ecological Research

Presentations by stream ecologists address a variety of organisms, physical characteristics, and temporal properties of stream systems. Despite the wide differences in research topics and background of speakers, four recurrent themes are evident. First, the need to understand the extent and causes of natural variation in biological communities is stressed as a prerequisite for developing effective management methods. Many studies have identified that populations, trophic organization, interspecific interactions, and species compositions are dynamic and change through time in stream systems. The magnitude of natural change is recognized as considerable and difficult to predict but not solely a random phenomenon. Second, the importance of natural variability and timing of flow events (especially floods and floodplain inundation) is recognized. Much of the variability in stream biota probably comes from the natural variability in streamflows. Research shows that organisms are adapted to seasonal cycles in stream discharge, and maintaining these cycles will be necessary to preserve natural system properties. Third, the critical importance of early life stages in determining adult population dynamics is discussed. Although several speakers identified this area as integral for explaining the natural dynamics of stream fish populations, they also agree that present knowledge of early life biology is poor and hampered by the continued emphasis on studies of adults and late juveniles. Finally, the complexity observed in warmwater communities and species–habitat relations is emphasized. Despite high complexity, the presentations include some generalized patterns (e.g., invertebrate production, faunal declines, fish–habitat organizations) that are useful for stream management. In general, recent research and probable future emphasis seem to center on relations among different types of aquatic habitats, causes of variance in the biological characteristics of streams, and the role of temporal habitat variability in communities.

Ecosystem Characteristics and Biological Productivity of Southeastern Coastal Plain Blackwater Rivers

by

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Blackwater rivers in the Coastal Plain of the southeastern United States are characterized by broad floodplain swamp forests and the presence of large amounts of woody debris (snags) in the main channel. Since they are located in a relatively flat landscape with low gradient, Coastal Plain streams are not as subject to impoundment as streams and rivers in higher gradients. However, snagging, channelization, and drainage of adjacent wetlands occur frequently in such systems, and all three can cause substantial ecological damage.

The Satilla and Ogeechee rivers are blackwater river systems in the southeastern Coastal Plain. They are the two largest river systems in Georgia that remain free from any major impoundments from their headwaters to the sea. Furthermore, there have not been major attempts to clear snags or channelize most of the upper portions of these rivers. However, the lower section of the Satilla was regularly cleared of snags until the early 1950's.

In both the Satilla and Ogeechee rivers, snags are the major site of invertebrate biomass, production, and diversity (Benke et al. 1984a, 1984b; Wallace and Benke 1984; Benke and Meyer 1988). River sediments generally have lower production and diversity, although they constitute larger areas. Many of the snag animals are filter-feeding aquatic insects (e.g., caddisflies, black flies, midges, mayflies). Other snag-dwellers include invertebrate predators such as dragonflies, stoneflies, and hellgrammites. Furthermore, the majority of drifting invertebrates (70–80% of numbers and biomass) originates from the snag habitat in the Satilla River (Benke et al. 1986). Also in the Satilla River, at least 50% of the prey of insectivorous fishes (e.g., bluegill, redbreast) originates from snags. Thus, invertebrate production—and hence probably much fish production—is directly related to the abundance of snags.

Interactions with the adjacent river swamps (increasing river width by as much as 40 times) are extremely important to animal productivity in these Coastal Plain rivers. The major organic foods for snag invertebrates seem to originate from swamps in the form of suspended soil bacteria or fine particulate organic matter (Edwards 1987; Edwards and Meyer 1987; Wallace et al. 1987). During flood periods, invertebrate production increases on the submerged swamp substrate and on wetted tree trunks. Inundation of the floodplain usually occurs for 3–4 months of the year and provides for a great expansion of feeding and spawning opportunities for many fish species.

Long-term management of Coastal Plain rivers should emphasize protection of the floodplain forests, especially by discouraging snag removal and channelization. In areas that have been largely cleared of snags or channelized, fish and wildlife managers should consider the introduction of woody debris to reestablish productive invertebrate and fish habitat.

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Biology and Ecology of Mollusks in Streams

by

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The indigenous freshwater mollusk fauna of the United States consists of about 765 species, including 500 species of snails, 40 species of fingernail clams, and 225 species of mussels. Two major subgroups of snails, pulmonates (lunged) and prosobranchs (gilled), have adapted to most freshwater environments. Pulmonate snails lack an operculum, respire via a pulmonary sac, are hermaphroditic (monoecious), and form light shells for a fairly active existence. In contrast, prosobranch snails have an operculum, respire through gills, have separate sexes, and form heavier shells for a more sedentary lifestyle. The 15 families of snails in freshwater rivers in the southeastern United States are dominated by the Pleuroceridae. These snails are small to medium in size and have distinctive solid, dextral shells, which are used for species identification. Pleurocerids are most abundant on cobbles and boulders in rocky shoals.

Freshwater fingernail clams (Pisidiidae) and the exotic asiatic clam (*Corbicula fluminea*) are small bivalves (<35 mm) that occupy both lotic and lentic habitats. Pisidiids are hermaphroditic, reproduce once or twice each year, and brood their young in marsupial sacs. The asiatic clam is also hermaphroditic, spawning twice a year and brooding embryos in its gills. It lives 2–3 years and achieves a maximum size of 35 mm in most rivers. Since 1924, this species has spread from the Pacific Northwest and now occurs in roughly 40 States. Because of its high reproductive potential, fast growth rate, short generation time, and high population densities, this exotic clam may compete with—and eventually displace—some of the indigenous bivalve fauna.

Of the 225 freshwater mussel species (Unionidae) in the United States, 34 are listed by the Federal Government as endangered; most of these occur in the Southeast, and many more species have been proposed for listing. The mussel reproductive cycle is similar for all species. During the spawning period, males release sperm into the water column. The sperm are then taken in by females during siphoning. Eggs are fertilized and incubated in the gills until larvae (glochidia) are mature. Glochidia are released into the water and must attach to the fins or gills of appropriate host fishes to encyst and metamorphose to the free-living juvenile stage. Juveniles are about 200 μm and require roughly 5 years to become sexually mature. Mussels are the longest-lived freshwater mollusks; individuals of some species have been aged at more than 50 years. The extreme longevity and the annual release of prodigious numbers (10,000–100,000) of glochidia per female provide the necessary reproductive potential to contact appropriate fish hosts and continue recruitment.

Habitat requirements for most unionids are fairly specific: clean, flowing water and a stable substratum of mixed particle sizes. In the Southeast, declines in the mussel fauna have been attributed principally to river impoundments, siltation, and water pollution. Numerous hydroelectric and flood control reservoirs have eliminated riverine conditions in most drainages, altering substrate and changing fish species composition. Poor land-use and mining practices have silted many streams and rivers, eliminating or reducing mussel populations in many river reaches. Point and nonpoint pollution have resulted in acute and chronic mortalities from herbicides, toxic spills, heavy metal discharges, sewage plant effluents, coal waste deposition, and many other anthropogenic activities. Over the last decade, an improvement in water quality in many rivers has occurred, which may allow the natural recolonization and recovery of some species. However, most endangered species are not likely to recover without the implementation of specific activities identified in recovery plans. Additional research and improvements in environmental quality

should provide opportunities to expedite the recovery of these unionids.

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Aquatic and Terrestrial Linkages: The Role of Floodplains

by

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Linkages between terrestrial and aquatic ecosystems occur at several orders of scale. At a landscape scale, stream basin characteristics control longitudinal and lateral (floodplain) interactions among aquatic habitats. The amount of organic matter input from the basin is dependent on the shape, recent use history, and nature of the floodplain. Lateral movement of organic matter has been severely affected in many river systems by levee districts, which have effectively blocked the stream from its valley. The nature and timing of organic matter input into lotic ecosystems will directly affect processing this material. Changes in the species composition of the riparian forest will affect the invertebrate assemblages in the stream because invertebrate life histories have evolved to respond to the resources that have historically been available. Changes in the invertebrate community can directly affect the fish assemblages. Removal of snag habitat has been a major cause in the decoupling of terrestrial and aquatic interactions. Debris dams serve as habitat for fish and provide sites for accumulation and subsequent processing of organic matter.

The closest link between terrestrial and aquatic ecosystems is bacteria. It has been shown that bacteria can only respond to natural sources of organic matter to which they have been previously exposed. This response is both physiologic and genetic. Populations of bacteria appear to be adapted to the organic matter found in the adjacent riparian and floodplain habitats. By changing the source of organic matter, the aquatic systems become less efficient in processing, and the greater portion of material is lost. As processing efficiency declines, there is less secondary production of invertebrates, which will lead to a decrease in fish and other vertebrates that rely on the invertebrate biomass.

More research is needed to determine how tight the linkage between terrestrial and aquatic systems is and how changes affect higher trophic levels. Clearly, management of floodplain and riparian forests will affect all trophic levels in stream systems, and it is important that we document these effects so that effective management practices can be developed.

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Ecology of Southeastern Stream Fishes: Geographic Macro- and Micro-habitat Considerations

by

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An understanding of natural variation in fish assemblage composition is important in developing and implementing stream management plans. Over the last decade, fish ecologists have worked to understand the patterns of organization and variation in communities and then to explore the mechanisms responsible for these patterns. Studies of stream fish assemblages necessarily emphasize current conditions and interactions, simply because we are faced with finite lifespans, careers, and budgets. While human and professional mortality are facts that we must deal with, our mortality—and the resulting short-term view of ecological communities—has led to problems of interpretation. I will identify three problems confronting fish community ecologists and suggest approaches to dealing with them.

The first problem is that ecologists often approach communities with the idea that biotic processes that structured such systems are still ongoing and are unaltered in the particular habitat under study—the fallacy of community stasis. It is important to have a temporal perspective on the subjects that we study as fish ecologists—namely species, assemblages, river systems, populations, and microhabitats. In terms of species, the age of most of the fish fauna in the southeastern United States probably dates from the late Miocene or early Pliocene. For instance, ancestral suckers, minnows, and darters are thought to have dispersed over a North American land route (Beringia) during the early Tertiary (Gilbert 1976), when such a connection between Europe and America still existed.

Somewhat later, the modern darter (tribe Etheostomatini) arose from an unknown percid ancestor. Vicariant events, such as isolation in many small streams and the disruption of highland areas, led to the evolution of numerous species of *Etheostoma* and *Percina* (Collette and Banarescue 1977). The minnows date from about the same age, with diversification of the largest genus, *Notropis*, probably occurring in the Mississippi Basin in the early to middle Pliocene (Gilbert 1964, for *Luxilus*). Fossil data indicate that centrarchids may date from the Oligocene (Gilbert 1976), and *Lepomis* occurred in the Miocene (Swift and Wing 1968). Thus, the three largest families of freshwater fishes have diverged since the Miocene or Pliocene (Gilbert 1964; Collette and Banarescue 1977), and therefore we are most concerned with recent, Pleistocene, and Pliocene geology in attempting to understand fish distributions and the history of interactions.

In dealing with the diverse fishes of the North American highlands, which include the area of greatest diversity of North American freshwater fishes, Mayden (1987a, 1987b) concluded that these faunas are older—probably much older—than the Pleistocene. While actual dates of origin of individual species are uncertain and will probably remain so, it is evident that the organisms we are studying have existed as species for 2–5 million years.

The message for ecologists studying stream fishes is clearly that these species are carrying considerable "historical baggage." Also, it is most realistic to expect that adaptations to reduce competition or predation should be generalized rather than specific to particular species or microhabitats. Many studies of fish resource partitioning attempt to interpret interspecific differences in resource use as due solely to local spatio-temporal interactions, ignoring the evolutionary history of species. The history of the southeastern fish assemblages tells us that each species in an assemblage represents a long, spatially integrated, and potentially independent evolutionary history, and at any given point a species may not even be well adapted to its environment (Jaksic 1981).

Whether or not assemblages vary significantly over time has been a major issue in the attempts to understand factors responsible for maintaining community structure (e.g., Grossman et al. 1982; Ross

et al. 1985, 1987; Matthews et al. 1988). The available literature on stream fishes indicates that persistence of assemblages may vary between harsh and benign habitats, but that values of well over a decade are not uncommon. However, the upper limits of temporal persistence are essentially all due to the length of the study period rather than an actual limit to persistence of the assemblage (Ross et al. 1987). The actual age of assemblages (considered from a persistence viewpoint) may range from a matter of years or decades—perhaps upwards to the age of river systems, on the order of 8,000–100,000 years. For instance, the drainage characteristics of many stream systems changed significantly during the Pliocene–Pleistocene transition due to glaciation, sediment transport, withdrawal of marine transgressions, and coastal uplift (Swift et al. 1986; Mayden 1987b). Thus, many systems date from the last 10,000 years.

As a further complication, all species pairs in a contemporary assemblage may (and probably do) differ in the lengths of their associations. Rarely should we expect that an assemblage, once formed, has existed unchanged over time. For instance, Mayden (1987a) points out that we need to distinguish between vicariant and dispersive events in the formation of fish assemblages. There are probably older core assemblages, due to historical vicariant events, that have periodically gained species from dispersive events. Thus, the degree of expected coevolution should vary, even within an assemblage.

The duration of individual populations is on a much shorter time scale. Most southeastern stream fishes have fairly short lifespans of 2–7 years. For instance, red shiners generally do not live beyond 3 years (Farringer et al. 1979), darters may live 3–4 years (Page 1983), topminnows may live 4–5 years (Fisher 1981), and centrarchids may live more than 10 years (Carlander 1977). Local microhabitats also may change on a scale from hours to years, often operating on temporal scales that approach or exceed the life span of individuals within a species. The temporal duration of microhabitats varies strongly with geographical region.

The second problem is the tacit assumption that present-day conditions of seemingly undisturbed systems represent the ecosystems in which the stream fauna evolved or, at least, the conditions that the assemblage has encountered for the majority of its history. In other words, we want to be able to make generalizations from our work that rest on the assumption that ours is a natural system, essentially unchanged from prehistoric times—the fallacy of the primeval stream. Historical changes of the last 200 years (such as widespread deforestation during the late 1800's and early 1900's) are documented well enough to argue strongly that there are virtually no southeastern streams that have escaped some degree of man-induced habitat change.

The third problem is less an outcome of our mortality than of our ability to ignore certain life history stages while emphasizing others; however, this problem is definitely related to time limitations—the fallacy of ontogenic stasis. In a review of resource partitioning studies of fish assemblages, Ross (1986) listed only one study that examined both larval and adult life history stages. Thus, most of what we know about fish resource use in general, and certainly of stream fish resource use, is from only one or two life history stages. A quick perusal of the issues of *Current Contents* from 1986 to the present, as well as searches of *Transactions of the American Fisheries Society* and *Copeia* for 1985–88, indicates that there is less emphasis on larval ecology of freshwater fishes than on larval ecology of marine fishes. It is safe to generalize that the life cycle and associated habitat use are not completely known for a majority of southeastern stream fishes. The critical question is whether the lack of knowledge of early life history stages significantly weakens our understanding of resource requirements of the fishes and any management plans based on those requirements. Available data indicate that an understanding of resource requirements of fishes and the subsequent management plans are substantially compromised if they do not include information from early life history stages.

In summary, finite careers and budgets encourage studies of communities and stream systems on short timescales and on life history stages that are most readily collected, observed, or identified. This has led to three basic errors: (1) ignoring historical effects in the structuring of communities; (2) assuming that present day studies deal with pristine systems; and (3) focusing on the most easily studied life history stages while ignoring stages that may, in fact, be more critical to an understanding of community function.

Further understanding of assemblage formation and maintenance requires a broader scope—one that considers ongoing as well as historical effects, and one that recognizes that both biotic and abiotic forces may act differently on the various life history stages.

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A Pluralistic View of Factors Influencing Warmwater Stream Fishes: Implications for Environmental Management

by

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Fish mortality is usually greatest during early life stages. Therefore, establishing what determines growth and mortality of fishes during early life stages will be essential for understanding fish population dynamics and for predicting the effect of environmental modifications on fishery resources. Unfortunately, our understanding of the influence of abiotic and biotic factors on early life stages of fishes in warmwater streams is poorly developed. Conducting controlled experiments is difficult and we lack a conceptual framework for identifying critical ecological processes.

In a previous descriptive study (Schlosser 1982a), I examined temporal and spatial variation in fish community structure along a pronounced habitat gradient. Based on these results I proposed a conceptual framework that attempted to integrate the relative effect of abiotic and biotic factors on fishes in small warmwater streams (Schlosser 1987a). From this conceptual framework it was evident that three factors potentially have major effects on growth and survival of early life stages of stream fishes: (1) harsh winter conditions, (2) direct effects of flow regime on invertebrate resource availability and fish survival, and (3) trophic interactions, including predation and competition.

My recent research used both descriptive and experimental approaches to examine the effects of trophic interactions and flow regimes on juvenile fishes. Results of this research indicated that (1) spring floods during spawning periods are associated with a major decrease in the density of juvenile cyprinid and centrarchid fishes (Schlosser 1985; see also Harvey 1987; Bain et al. 1988); (2) elevated (nonscouring) flow results in a dramatic increase in invertebrate abundance during spring and summer, when most growth of juvenile fishes is likely to occur (Schlosser and Ebel 1989); (3) in the absence of predators, small fishes depress invertebrate abundance in pools but not in riffles (Schlosser and Ebel 1989); and (4) centrarchid predators restrict juvenile fishes to shallow refugia while engaging in both species- and size-selective predation (Scholosser 1987b, 1988b).

These results indicate that multiple abiotic (e.g., channel morphology, flow regime) and biotic (e.g., resource depression, predation) factors interact to influence growth and survival of early life stages of stream fishes. Because many environmental modifications simultaneously alter several of these variables, such modifications are likely to have considerable effects on the fishery resources in modified streams. For example, agricultural land use alters flow regime, channel morphology, seasonal timing of resource availability, and abundance of piscivorous fishes (Karr and Schlosser 1978; Schlosser 1982b). These modifications, in turn, have major effects on temporal variability and juvenile recruitment in headwater and downstream areas (Schlosser 1982b).

These results also indicate that the development of a predictive paradigm for environmental and fisheries management in streams will require refinement in our understanding of how abiotic and biotic factors interact to affect early life stages of stream fishes. This refinement will be achieved through a combination of (1) long-term monitoring to establish the influence of natural environmental variability on growth and survival of early life stages and (2) experimental studies to assess the effect of variability in flow and temperature regime on trophic interactions. Lastly, fishes are highly migratory during reproductive activities, and stream systems are heterogeneous and interconnected. Hence, it will be necessary to explore the effects environmental modification in one stream have on population and community dynamics of fishes in adjacent parts of the stream system (landscape approach of Forman and Godron 1986).

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Fish Community Structure and Stability in Warmwater Midwestern Streams

by

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Impact assessment generally implies a search for ways to measure and predict changes in stream systems that ensue from cultural alterations. In order to really know the degree to which a cultural activity results in change in stream systems, something must be known of the magnitude of change or variation that occurs naturally. Even if managers are constrained from long-term studies of each stream of interest, the long-term data sets that do exist can be used to predict the kinds and degree of changes that may naturally occur in a variety of stream systems.

Over the last two decades, a relatively long-term data base has been accumulated from our collecting activities assessing fish assemblages and faunas in three midwestern streams: Piney Creek, Izard County, Arkansas; Brier Creek, Marshall County, Oklahoma; and Kiamichi River, southeastern Oklahoma (Ross et al. 1985; Matthews 1986; Matthews et al. 1988). These data sets have been evaluated at two levels: stability of whole-stream faunas (based on pooled collection data at multiple sites on each stream) and stability of fish assemblages at each individual site over time. The data set spans 1969–86 for Brier Creek, 1972–86 for Piney Creek, and 1981–86 for Kiamichi River. Each of these time intervals allows for one to two complete generations of at least the small fish species that are most abundant.

The whole-stream faunas of all three streams have been highly persistent (in terms of presence and absence of species) and stable (based on abundance data) in all years of study. No common species has been lost from any of the streams, and no "rare" species has become common at any of the sites. Both similarity indices (Morisita's and percent similarity) and statistical rank correlation of species abundances show that all three streams have been stable, and that the most environmentally stable stream (Piney Creek of the Ozark uplands) has had the greatest degree of fish population stability across all years.

At individual sites on each of the streams, the local fish assemblage has had a tendency to be stable, although exceptions exist in all three systems. Where individual collecting sites have shown a marked change in the composition of the fish assemblage, this change can usually be attributed to known events, such as deposition of large woody debris by floods, death of fish in intermittent headwaters due to drought, or other such disturbances. However, recolonization of intermittent headwaters is not a random phenomenon. Matthews (1987) showed that existence and persistence of fish species in harsh headwaters of at least one of the streams was directly related to their ability to withstand low oxygen conditions, which are common. Thus, there is both a chance component and a directed component to dynamics of assemblage composition in headwater streams.

The long-term study of Piney Creek included a flood event that was clearly of 100-year magnitude. Vertical water levels were elevated as much as 12 m at some sites, and physical destruction and disturbance of habitat along the creek was extreme. Immediately after the flood, fish assemblages showed some differences from those that existed before the flood, but after 8 months, both the whole-stream fauna and most local assemblages were statistically indistinguishable from their composition before the flood. This well-documented event—and other accounts of flood effects on fish in midwestern streams (Gelwick 1990; Harvey 1987)—indicates that while floods may have an immediate effect on composition of midwestern fish communities, these changes are largely transient, with the systems returning to preflood conditions rather rapidly.

Some methodological recommendations are possible from collections made on these streams. There is a longstanding question in fish community assess-

ment as to what length of stream constitutes a most appropriate and cost-effective collecting activity. In many standardized procedures, a reach of 100 m has been adopted for small- to medium-sized streams. In contrast, my collections at a site have always included a minimum of 200 m—often up to 400 m—of stream, in order to sample sufficient distance to find representative habitat for all species that are present. In 1988, I compared fish collections in adjacent 100-m sections of Brier Creek at a total of 5 sites, or 10 segments. Typically, the overall fish assemblage in spatially adjacent 100-m segments had percent similarity in composition of only 60–70%, and hence a different picture of the fish community resulted if a total of 200 m was included in the sample instead of only 100 m. Accordingly, in at least some midwestern streams, I suggest that more than 100 m of stream reach should be sampled, whether the goal of the study is assessment of some particular effect or the acquisition of long-term data on basic ecological properties of the fish community.

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Centrarchid–Habitat Associations in Ozark Streams

by

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Any goal of sportfish management should include satisfactory sustained angling benefits, which can occur only by maintaining satisfactory habitat and balanced fish populations and communities. Appropriate habitat is an important prerequisite for balanced fish populations in streams. Yet, not enough is known about habitat requirements of most fish. Unless the stream is highly degraded, it is not possible to take a generalized habitat model developed from the literature and apply it to a particular stream with much hope of identifying ways to increase fishing quality. Each stream, or at least each stream type, must be considered individually because fish are often plastic in their behavior and requirements, and a limiting variable in one geographic area may be unimportant in another.

Two approaches have been taken by researchers working on species–habitat relations. The first is to determine how individual fish use particular habitat elements and then infer something about the entire population. This is dangerous because there is no known empirical linkage between the two. Drawing ecological meaning from data on habitat use or preference is similar to trying to evaluate mechanisms from correlations. Habitats of a stream fish are often measured in terms of either absolute or relative use, yet these data reveal nothing about how important a particular habitat variable might be nor how the variables are interrelated.

A second approach is to correlate some population characteristics, generally density or biomass, with some habitat feature. While some studies are successful in relating fish quantity to some habitat quantity, their usefulness in a management context is limited. There is generally no consistent relation between fish amounts and habitat amounts. Although a biologist can often distinguish a "good" fishing stream from a "bad" one, unless a stream is initially severely degraded there is little evidence that altering a particular habitat variable will cause a predictable corresponding change in the fish population. We have little understanding of factors affecting carrying capacity of a warmwater stream—especially the relation between food and habitat. While our goal is to quantify critical factors of the stream environment, it does not appear that this goal can be attained in the near future. What we as fishery researchers should do is isolate a subset of environmental variables, pertinent to the particular situation at hand, from a larger set of variables considered generally important to a species—such as those listed in the U.S. Fish and Wildlife Service's Habitat Suitability Index Models. If done carefully, we should be able to pinpoint particularly important habitat variables that influence the success of a population and on which management efforts can be focused.

Habitat conditions can be related to a life stage in one of three ways. Nonuseful habitat conditions are those not required for the population's survival and may have an adverse relation to a population (e.g., extremely high water velocity). Useful habitat conditions provide a necessary element for the population's well-being when other habitats also provide the same elements (e.g., rootwads and log jams). Essential habitats are those required for the population's well-being.

We must determine which habitat elements fit into which category to effectively manage a species. Unfortunately, nonuseful habitats are often impossible to identify because even if an area is unused by fish it may still be important as a food-source area or barrier to predators or competitors. Useful habitats can be detailed if the species is studied in more than one location. Essential habitat features may or may not be limiting to a population.

Based on the preceding concepts, I evaluated the habitat relations of Ozark stream centrarchids, with emphasis on smallmouth bass (*Micropterus dolomieui*) and lesser emphasis on rock bass (*Ambloplites*

rupestris) and longear sunfish (*Lepomis megalotis*). I studied behavior of individual fish by underwater observation and telemetry; documented population characteristics of density, biomass, and condition; and attempted to relate habitat use by individuals to the well-being of the population.

I documented daytime habitat use in summer using underwater observation. After I corrected for differences in the amount of open water and each cover type observed, over 90% of individuals of each species were found to be positioned within 1 m of cover. Overlaps in cover use between species suggested that cover is not important in interspecific segregation. Species segregation was strongest by current velocity, with smallmouth bass using cover in the fastest water and longear sunfish using cover in the slowest water. Current velocity was also important to smallmouth bass and longear sunfish as an element of intraspecific segregation. Strong positive linear relations between water depth and fish size were noted for all three species, but water depth was not important in interspecific segregation. Substrate use appeared to be dependent on cover use, and results were inconclusive. All three species shifted habitat throughout the day, but the greatest shifts occurred at night.

Telemetry studies on smallmouth bass substantiated the observational studies and added information on fish habitat associations in winter, when fish used boulders almost exclusively. Both studies showed that smallmouth bass avoided shallow areas and existed almost exclusively in current velocities <0.2 m/s. I believe that the results of studies emphasizing individual fish can be used with other studies to delineate essential habitat conditions for smallmouth bass populations.

Regression models from population studies showed that depth and current did not correlate with standing crop of smallmouth bass. Perhaps these variables are important but are available in greater amounts than needed. What the regression model did show is that boulders had a positive relation to standing crops. Perhaps boulders are an essential habitat or a limiting factor for smallmouth bass in these streams because of some important, yet still unclear, function during cold-water periods. Fish may be more energetically efficient when associated with boulders during stressful periods. The significant, positive, but weak relation between boulders and fish density or biomass indicates that fish use boulder areas in a nonlinear fashion. Underwater observations indicated that a single boulder properly situated could accommodate as many as eight adult smallmouth bass. Thus, while a particular aspect of the habitat may be essential, its quality may be far more important than its quantity. These studies suggest that obtaining the information necessary for effective management of a fish population requires several approaches, and all these approaches have to be repeated for different life stages.

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Fish Larvae—Ecologically Distinct Organisms[1]

by

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The behaviors and habitat requirements of larvae of most warmwater stream fishes are distinctly different from those of their juvenile and adult counterparts (Braum 1978; Marcy et al. 1980; Snyder 1983, 1985; Floyd et al. 1984; Faber 1985). The larval period—the interval of fish development from hatching or birth to loss of finfolds and development of adult complement of fin spines and rays—entails many and often dramatic changes in morphology and physiology. Also, it usually covers a number of short-term shifts in lifestyle and habitat. Indeed, a fish larva is often ecologically distinct not only from its juvenile and adult counterparts but from itself during certain intervals within the larval period.

The initial habitat of most fish larvae is the spawning ground. For warmwater stream fishes and anadromous species, there is much diversity in the earliest habitat requirements and behavior of larvae. Initial larval habitats, associated environmental requirements, and behaviors are considered in a classification of reproductive guilds by Balon (1981, 1984). The 22 families and over 260 species (Swift et al. 1986) of freshwater and anadromous fishes in the southeastern United States represent at least 17 of Balon's reproductive guilds (Table).

Some fishes remain on or near the spawning grounds throughout much or all of their early development, and some may use the same habitat throughout their life cycle. However, for most stream fishes, the initial habitat rapidly becomes inadequate, and the larvae either drift or actively migrate to more suitable nursery grounds, sometimes hundreds of kilometers downstream. The larvae and early juveniles of most stream fishes tend to use nearshore areas with relatively slow-moving water near cover, vegetation, or sharp vertical relief.

Larval fish size largely dictates the characteristics of suitable nursery habitat and the role larvae play in the aquatic system. In southeastern fresh waters, fish larvae may be as small as 2 mm in total length (TL) at hatching (e.g., white crappie [*Pomoxis annularis*]) and early larvae of some (e.g., striped bass [*Morone saxatilis*], freshwater drum [*Aplodinotus grunniens*], and emerald shiner [*Notropis atherinoides*]) inhabit pelagic waters and constitute a part of the planktonic community. These early pelagic larvae may be food for predatory copepods, whereas later larvae of these fishes may reverse the roles and prey on the copepods. Most fish larvae feed on rotifers and other small zooplankton; however, some are piscivorous and may consume other fish larvae nearly as long as themselves. Clark and Pearson (1979) observed fish larvae in the stomachs of over 25% of small (4–5 mm standard length [SL]) freshwater drum larvae. For this species, piscivory decreased with size and was not observed in larvae or early juveniles over 9 mm SL.

For most stream fishes, extremely high mortality during the embryonic and larval period is normal and accommodated by their reproductive strategy. Environmental effects that substantially add to, or reduce, natural larval fish mortality can have a corresponding effect on the eventual size of the adult population. The effects of changes in habitat and community structure on fish larvae can be quite different from those on juveniles and adults. The most common effect is loss or alteration of larval fish habitat. Fish larvae can be particularly sensitive to physical and chemical water pollution. Biotic alterations that might not be expected to have an adverse effect on native fishes, such as the introduction of exotic forage species, might indeed affect native populations by excess predation on, or competition with, their larvae. Fish larvae of many species are especially vulnerable to entrainment in water withdrawal systems for irrigation, domestic and industrial water supplies, and power plant cooling.

[1]Contribution 42 of the Larval Fish Laboratory, Colorado State University.

Table. *Reproductive guilds*[a] *by family for fishes in fresh waters of the southeastern United States.*

		Nonguarders								Guarders						Bearers		
	Number of guilds	Open substrate						Brood hiders		Substrate choosers		Nesters				External	Internal	
Family		A11	A12	A13	A14	A15	A16	A23	A24	B13	B14	B22	B23	B25	B27	C14	C22	C24
Acipenseridae (sturgeons)	2		X[b]	X														
Polyodontidae (paddlefish)	1		X[b]															
Lepisosteidae (gars)	1					X												
Amiidae (bowfin)	1													X				
Clupeidae (herrings)	3	X	X		X													
Hiodontidae (mooneyes)	1		X															
Salmonidae (trouts)	1							X										
Umbridae (mudminnows)	1													X				
Esocidae (pikes)	1					X												
Cyprinidae (minnows)	9	X		X	X	X	X	X	X				X		X			
Catostomidae (suckers)	3			X		X	X											
Ictaluridae (catfishes)	2												X		X			
Amblyopsidae (cavefishes)	1															X		
Aphredoderidae (pirate perch)	1											?				?		
Cyprinodontidae (killifishes)	3				X	X					X							
Poeciliidae (livebearers)	2																X	X
Atherinidae (silversides)	1				X													
Percichthyidae (temperate basses)	2	X			X													
Centrarchidae (sunfishes)	4									X		X	X	X				
Percidae (perches)	7		X		X	X	X	X						X	X			
Sciaenidae (drum)	1	X																
Cottidae (sculpins)	1														X			
Number of families		4	5	3	6	6	3	3	1	1	1	1(2)	3	4	4	1(2)	1	1

[a]Guild codes: A11 = pelagic spawners; A12 = rock and gravel spawners with pelagic larvae; A13 = rock and gravel spawners with benthic larvae; A14 = nonobligatory plant spawners; A15 = obligatory plant spawners; A16 = sand spawner; A23 = rock and gravel hiders; A24 = cavity hiders; B13 = rock and gravel tenders; B14 = plant tenders; B22 = miscellaneous substrate nesters; B23 = rock and gravel nesters; B25 = plant material nesters; B27 = hole nesters; C14 = gill-chamber brooders; C22 = obligate lecithotrophic livebearers; C24 = viviparous trophoderms (see Balon 1975, 1981, 1984 for detailed descriptions of these and other guilds).

[b]Between A12 and A13—recently hatched larvae drift near bottom (epibenthic drifters?).

In our attempt to evaluate environmental impacts and aquatic management programs, we often turn to models, habitat-quality indices, and related tools. Unfortunately, these models or tools can be no better than the data on which they are based, and for most species of fish, reproductive and early life history data are often lacking, inadequate, or questionable. For southeastern fishes and aquatic systems, there is much basic research that needs to be done to fill the immense void in our knowledge, including such basic tasks as determining what the larvae of many fishes look like. In the Southeast, we have only four limited manuals for larval fish identification (Hogue et al. 1976; McGowan 1984; Conrow and Zale 1985), although some species are covered in taxonomic manuals for other parts of the country (Auer 1982; see lists in Snyder 1983 or Simon 1986 for others). The importance of accurate identification of specimens cannot be overemphasized because critical resource management decisions are sometimes based on species-specific field data. Before we can effectively proceed with field studies on fish larvae in the Southeast, we must be able to accurately identify specimens.

The early life stages of fishes must be an important concern in the development of management plans and impact assessment methods. Aquatic ecologists and fishery biologists are often too preoccupied with the needs of adult fishes to recognize the differing requirements of earlier life stages. It is simpler and less costly to concentrate on one life stage of a target species than on a whole series of ecologically distinct stages; however, fish populations depend on adequate survival of their embryos, larvae, and early juveniles.

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Assessment and Applied Research

The presentations by biologists involved in developing or using stream assessment methods reflect the challenges and tradeoffs of simplifying complex systems for use in practical but valid methods. The speakers of this group vary considerably in their approaches, although all are primarily interested in stream fishes. One group of speakers emphasizes the need to develop and use methods based on measures of fish communities and natural flow regimes. These biologists believe that assessment methods need to use information on all fishes rather than a few target species, and they address patterns of streamflow rather than simply minimum flow. Other speakers emphasize the practical aspects of stream assessment and the usefulness of assessment methods based on easily estimated and narrowly focused measures. These biologists deal with stream system complexity and practical constraints by advocating the use of methods that vary from simple statistical standards (e.g., median August flow, percent of mean annual flow) to computer modeling. Ironically, when the management outcomes of different assessment methods were compared (see Orth and Leonard), the differences were not that striking. The solution to appropriately simplifying the complexities of stream communities seems to be far away but may emerge from the choice of methods made by management biologists as they face future stream impact issues.

The IBI: A Quantitative, Easily Communicated Assessment of the Health and Complexity of Entire Fish Communities[1]

by

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Two basic questions face biologists interested in assessing and predicting impacts on stream ecosystems: Why do we need assessments of biological communities? How can data on biological communities (species and abundances) be made useful for nonbiologists (decision-makers and the public)?

Answers to these questions are central to implementing the Clean Water Act. Although the act requires restoration and maintenance of biological integrity (Section 101) and biological assessment (Sections 105, 303, 304, 305, 404), *biological integrity* remains poorly defined and rarely measured. States are required to establish water quality standards consisting of designated biological uses (broad goals) and criteria (which if met are presumed to protect the uses); however, both the uses and the national chemical criteria are inaccurate. Uses such as *aquatic life* and *warmwater fish* are so broad as to be met by any form of aquatic life or species of fish, in any abundance.

Nationally established chemical criteria ignore naturally occurring differences in conventional water quality parameters, such as dissolved oxygen, pH, and the ionic character of waters. The ionic character of waters is particularly important because it affects the activity of toxic chemicals. Equally important, single-chemical criteria do not consider combined-chemical effects, which may mitigate or magnify results. Single-species toxicity tests may not be appropriate or sensitive measures of ecosystem effects because test species may not be representative of indigenous species or the most sensitive species and because populations can be devastated by indirect effects on competitors, predators, and prey. Furthermore, criteria exist for only a small number of chemicals. It is too expensive to develop specific criteria for every site, and it would create immense regulatory problems if they were developed. Water quality monitoring data are difficult to interpret because of the lack of an appropriate sampling and reporting framework and because of the weak link between water quality and ecosystem structure and function. Finally, there are many other factors besides toxics that may limit attainment of biological integrity.

Like toxics, physical habitat conditions are also surrogate measures of the communities that agencies are mandated to protect. Although physical and chemical conditions are necessary for understanding and explaining biological conditions, biological monitoring is preferable in many cases because it provides a direct assessment of the biological community. Moreover, the biota reflect the integrated chemical and physical quality of an area, and they often can be assessed at lower cost than physicochemical habitat parameters, such as priority pollutants and species-specific habitat. Millions of dollars have been spent on chemical monitoring and on the development of instream flow models; still, there is only limited quantitative information regarding the health of the resident biota at those sites. In the southeastern United States (unlike the West, where the methodology developed), meeting instream flow requirements for game fish or a small number of nongame species is underprotective of the entire fish assemblage because (1) most game species have wider habitat–flow tolerances (e.g., centrarchids vs. salmonids), (2) many nongame species have narrower habitat tolerances than the game species,

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and (3) the fish fauna has far greater species richness than in the West.

Direct biological monitoring of reference sites and sites of interest, conducted within an appropriately scaled ecoregion framework, can provide both an assessment of status and a prediction of effects, given similar stressors in similar ecoregions (Hughes et al. 1986; Omernik 1987; Hughes and Larsen 1988). Such information is generally more convincing to the public and to judges than are violations of chemical criteria and flow standards (E. Bender, USEPA Office of Water Enforcement and Permits, Washington, DC, personal communication). To increase the accuracy of our predictions, we must combine long-term biological monitoring and experimental field studies of the effects of key physical and chemical variables on populations and communities.

How can data on species and their abundances be made understandable to the public and to water resource regulators while retaining the ecological information that is meaningful to biologists? A recent attempt to communicate such information is the Index of Biotic Integrity, or IBI (Karr 1981; Karr et al. 1986). The IBI is a means of quantifying ichthyologists' judgments of the relative quality of a fish assemblage. It is based on a sample of the entire fish assemblage, not just game fish. The index incorporates professional judgment of fish assemblage health in 12 metrics and their scoring criteria, which are based on regional ideals. These regional standards are determined from historical data and data from minimally affected sites that characterize the region. The individual metrics differ in their range of sensitivity for detecting perturbations, and a degree of redundance is built into the IBI because no single metric can reliably indicate integrity. The metrics and variations on them (Karr et al. 1986; Miller et al. 1988; Plafkin et al. 1989) are summarized below.

Number of native fish species
Number of darter and benthic species
Number of sunfish and water column species
Number of sucker and long-lived species
Number of intolerant species
Percent top carnivore individuals
Percent insectivorous individuals
Percent omnivorous individuals
Percent tolerant individuals
Total number of individuals
Percent hybrid and exotic individuals
Percent diseased individuals

Five of Karr's original metrics have frequently required modification for streams outside the Midwest (Miller et al. 1988). It may be instructive to examine this process if IBI modification is needed in the southeastern United States. The numbers of benthic insectivorous species and sculpin species were used in New England and Oregon, respectively, where darters are depauperate or rare. Both substitutions contain species that use benthic habitats for reproduction and feeding, have small home ranges, and are sensitive to degradation. In the Southeast, madtoms might be added to this metric. The number of native minnow or water-column species replaced sunfish species as a measure of pool quality in Oregon and the Northeast, respectively. The species in both groups are sensitive to changes in pool or water-column habitats, and sunfishes are introduced or depauperate in both regions. Insectivorous cyprinids are a dominant trophic group in the Midwest, but where they are less dominant, Karr's original metric was changed to "all insectivores" or to "specialized invertebrate feeders." The proportion of individuals as green sunfish has been replaced frequently. This metric assesses the degree to which species that are tolerant to a variety of stressors dominate. Common carp, white sucker, creek chub (Miller et al. 1988), and the percent of individuals of the 12 most tolerant species (Ohio Environmental Protection Agency 1988) have all been used where green sunfish were inappropriate or insensitive. The percent of hybrids has also been problematic in several regions; introduced or exotic species were substituted in Oregon and Colorado. This modification may be appropriate in Florida also, as a measure of the breakdown of reproductive separation zoogeographically or simply as an indication of biological pollution.

There is growing interest in applying the IBI to water resource management. The IBI is presently being used by the Illinois Environmental Protection Agency, the Kentucky Cabinet for Natural Resources and Environmental Protection, the Ohio Environmental Protection Agency, and the Tennessee Valley Authority. Several other State agencies (Alabama Department of Environmental Management, Iowa Conservation Commission, Kansas Department of Wildlife and Parks, Nebraska Department of Environmental Control, New York Department of Environmental Conservation, North Carolina Division of Environmental Management, Oklahoma Department of Health, Vermont Department of Environmental Conservation, and Wisconsin Department of Natural Resources) and the National Park Service are testing the IBI for use as a monitoring tool. Ohio Environmental Protection Agency (1988) uses the IBI as a legal criterion in its water quality standards program, and the USEPA supports the

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Comparison of Instream Flow Methods for Western Virginia

by

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Reliable methods are needed for recommending minimum instream flows to protect aquatic life because conflicts between instream and offstream uses of water will increase in the future, even in some water-rich regions. Water resource planning can help avoid these conflicts only if we know the acceptable magnitude and duration of low flows to maintain current instream values. For basinwide planning purposes, simple methods, which require little or no field investigations, are required. While some of the existing methods tend to be conservative, the degree to which they protect habitat for fish is seldom determined. We applied physical microhabitat models (PHABSIM) for nine target fish species in four streams in the upper James River basin, Virginia, to (1) identify optimum flows to protect the fish fauna, (2) investigate the relation between optimum flow and average discharge, and (3) compare our findings with recommendations based on simple discharge methods.

Microhabitat availability for riffle-dependent species was most limited at low flows, whereas microhabitat availability for pool-dependent species was most limited at high flows. At each study site there was a rapid increase in riffle habitat as discharge increased above zero. Optimum flow maximized habitat for the most critically habitat-limited species or life stages. The recommended optimum flows increased with increases in stream size, but the slope was not constant; as stream size increased, a lower proportion of average discharge provided optimum habitat. The recommended flows were related to the average discharge (AD) with a power function: optimum flow (m^3/s) = $0.583\ AD^{0.746}$.

Water resource managers can develop flow recommendations for other streams in the upper James River basin based on the average discharge—assuming water quality is not a limiting factor. Aquatic Base Flow recommendations (i.e., September median flow) provided varying—but a reasonable degree of—habitat protection. The Montana method (10% average discharge) recommendations correctly identified degraded or poor habitat conditions, and the 30% recommendations corresponded to near optimum habitat in small streams but greater than optimum flow at the large-stream site. Seven-day, 1-in-10-year low flows (7Q10) provided very limited amounts of physical habitat for riffle-dwelling fishes. The results of our study provide a basis for making preliminary flow recommendations in this region from readily available data. Studies will be needed, however, to test the generality of the findings in other basins and the assumptions of the methods employed.

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Minimum Flow is a Myth

by

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During the warmwater stream symposium in 1980, when discussing low flow as a limiting factor in warmwater streams, I warned fisheries management personnel that the concept of a single minimum or base flow for fishery habitat maintenance that has evolved in the western region of the United States could very well become a real threat to low gradient, eastern, warmwater stream fisheries (Stalnaker 1981). The minimum flow concept rose from western water law as a mechanism to either reserve an amount of water from future appropriations or as a means of granting an instream water right for fishery purposes. This led to the myth that a consistent methodology could be used to establish a single minimum discharge value for any given stream. Experience has shown that as water becomes fully appropriated to upstream use or storage, the minimum flow, if not frequently violated in time, tends to become the average flow condition. Too often the minimum becomes the objective rather than the means to achieve some riverine fishery or recreation management goal. Such persistent low flows are not necessarily desirable from the water management perspective, being inflexible in the face of competitive uses or during unusual water supply conditions (e.g., drought), and certainly do not meet all the desired environmental needs. This difficulty with minimum flows arises in part because all the instream uses for which flows may be needed are not identified. Most often overlooked are necessary periodic high flows that move bedload, flush sediments, rejuvenate the floodplain, and generally maintain the structural characteristics of a stream channel, which should be maintained in dynamic equilibrium with its watershed (Stalnaker 1979).

A common misconception among water management personnel and consumers is that inclusion of all the identifiable instream uses of water in an instream flow requirement will dictate an additive treatment of their respective needs. This, it is further assumed, will result in total allocation of the stream flow to instream uses. Contrary to this view, a considerable degree of compatibility exists among many instream uses and downstream delivery requirements for offstream or consumptive uses. However, in order to deal with these compatible uses, the instream flow advocate and the water resource manager must be aware of both the timing and the magnitude of all the demands being placed on the stream system. Such a common understanding, which should lead to the identification of instream flow requirements, will protect all complimentary uses as well as meet downstream delivery requirements.

It is evident from reviewing the literature and from the discussions during this workshop that many methods for evaluating instream flow needs have evolved since the 1960's. I prefer to categorize such methods as "standard setting" or "incremental." Standard setting methodologies, on one hand, refer to those measurements and interpretive techniques designed to generate a flow value (or values) intended to maintain the fishery or recreational use at some acceptable level (usually dictated by policy). Incremental methodologies, on the other hand, are organized and repeatable processes by which (1) a fishery habitat–stream flow relation and the hydrology of the stream are transformed into a baseline habitat time series, (2) proposed water management alternatives are simulated and compared with the baseline, and (3) project operating rules are negotiated.

Trihey and Stalnaker (1985) suggested that a hierarchical approach to hydro licensing and relicensing be followed that in essence takes advantage of both the standard setting and incremental approaches. A three-tiered hierarchy was suggested including reconnaissance, feasibility, and operational or design studies for evaluating hydro projects. It is important to recognize that such licensing is

generally a multiyear process. Adopting the suggested hierarchical approach can lead to greater understanding among the resource agencies, the applicant, and the general public, leading to negotiated conditions for the license. Specifically, the reconnaissance study identifies the stream segments of potential impact, the project location configuration, and possible operating scheme. With- and without-project hydrologic conditions are compared to determine whether the project seems to be "benign" and compatible with resource agency policies. In other words, there is little change in the flow pattern below the project. In the feasibility study, the use of a previously set standard can be quite advantageous. At this level of analysis, comparison is made between the projected stream flow conditions and the stream flow maintenance standard to identify major issues and periods of incompatibility. Standard setting methods (such as the New England Flow Method and the Arkansas and North Carolina methods) were discussed during this workshop; they and the optimum flow proposed for western Virginia are excellent examples by which one can screen for hydro projects that seem to be incompatible with agency policy and environmental protection goals. When it becomes obvious that project operations and the maintenance of stream flow standards are incompatible, impacts need to be quantified and mitigation measures agreed on. Then much more detailed operational level studies are appropriate. Only during this third study phase do the incremental methods become useful and, in fact, necessary.

The majority of States now recognize instream flows and have identified procedures for incorporating such uses in water planning (Reiser et al. 1989). Adoption of a standard setting approach by the State Water Resources and Fisheries Management agencies greatly facilitates identification of incompatible water development projects during feasibility studies. Stream flow assessment methods, such as the Instream Flow Incremental Methodology used by the U.S. Fish and Wildlife Service, have consequently evolved to become environmental assessment techniques and are used for evaluating the effects of proposed reservoir construction, water diversions, or hydroelectric operations on downstream fish habitats. Quite often such impact assessments become a matter of comparison among several possible, but not always measurable, water management schemes, leading to the necessity of simulation modeling for making these comparisons. Only the physical–chemical aspects of the habitat are evaluated, and comparisons are judged on the potential habitat limitations that may result from a proposed change in the way stream flows are controlled and routed through stream segments. It is important to realize that minimum flows, optimal flows, and even stream flow standards are not impact assessment tools. When it comes to relicensing of hydroelectric projects, the questions really are focused on the effects that may result from a change in project operations. Minimum flow has no logical argument in such an institutional process and, in fact, as hydro projects go to increased peaking operations (involving daily and hourly rapid fluctuations in the tailwater releases), it is often the high flows that are of more concern from a biological standpoint than the low or minimum flows.

The challenge now before us is to progress beyond the minimum flow and even habitat impact assessment and to focus on scientific principles in understanding riverine systems. Management biologists must get involved with water management in riverine environments. By definition, management is a designed and directed change in a system. The improvement of basic understanding of ecology of our stream systems, coupled with the use of engineering tools and simulation modeling, provides an opportunity for fisheries to be enhanced downstream of the many hydroelectric projects coming up for relicensing in the 1990's. This will occur only if fishery managers and natural resource agencies do the designing and directing of the change in the operating systems, working hand-in-hand with the hydro project applicants and the Federal Energy Regulatory Commission.

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Considerations in Applying IFIM to Warmwater Streams[1]

by

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Increased diversion of riverine flows in the southeastern United States for irrigation, municipal, and industrial water supply has resulted in an increased awareness of the importance of adequate instream flows for protecting and maintaining riverine fishes. One of the most commonly employed means of assessing the effects of river water diversion on warmwater streams is the U.S. Fish and Wildlife Service's Instream Flow Incremental Methodology (IFIM). Because the methodology was originally developed for coldwater streams, a number of factors should be considered in applying the IFIM to warmwater streams. Some primary considerations include differences in hydrology, channel geometry, and biology between coldwater and warmwater systems.

The hydrograph of coldwater streams used to develop the IFIM is heavily influenced by snowmelt or groundwater recharge, which tends to result in long-term gradually varying flow conditions. In contrast, warmwater streams are more influenced by rain runoff and can, therefore, exhibit complex hydrologic patterns that reflect daily, synoptic, or seasonal influences. Consequently, simple hydrologic variables, such as mean or median monthly flow (commonly used to describe coldwater streams), may not be as relevant to aquatic biota in warmwater systems. Indeed, variability in the hydrograph and concomitant variations in habitat may be important features of these ecosystems, required for organic matter transport, nutrient cycling, or successful completion of life stages.

Channel morphometry in the coldwater stream systems used to develop the IFIM was usually simple, composed primarily of pool–riffle complexes, and variety in habitat conditions (depth, velocity, and substrate) was more limited than in warmwater systems. In contrast, many warmwater streams have compound channels and may include backwaters, sloughs, braided channels, snags, and other channel features that make habitat conditions within these systems more difficult to describe using cross sections and relatively simple hydrologic methods.

The biology of many coldwater biota (primarily salmonids) is well known and documented. Many studies have been performed that describe salmonid habitat requirements in terms of substrate and channel flow conditions. In contrast, the biology of warmwater aquatic biota is less known and often undocumented in terms of variables used in instream flow studies. In fact, even the term *warmwater biota* may be misleading because it implies the existence of a distinct category of aquatic biota restricted to warmwater stream ecosystems. In reality, there may be any number of warmwater "biotic assemblages."

Coldwater and warmwater stream ecosystems differ substantially in species number. Small- and medium-sized coldwater streams are usually characterized by a relatively small number (1 to 10) of fish species and usually include only one or two sport fishes. It is usually not difficult to identify a target species or life stage to serve as the focal point of the study. In contrast, a warmwater stream of similar size may have 30 or 40 species of fishes, with several of them having commercial or sport-fishing importance. Consequently, it is often difficult to identify a suitable target species for the analysis. If the habitat requirements of a large number of species are evaluated, then assessment of impact is complicated because each life stage may have substantially different flow optima.

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[1]The conclusions and interpretations presented in this paper were based on research conducted under the auspices of the Environmental and Water Quality Operations Studies and Environmental Impact Research Program of the U.S. Army Corps of Engineers by the Waterways Experiment Station. Permission was granted by the Chief of Engineers to publish this information.

Comparison of suitability curves for coldwater and warmwater fishes reveals that, generally, warmwater stream biota are characteristically habitat generalists, able to occupy a wider range of depth, velocity, and substrate conditions than coldwater biota. Consequently, an analysis using the Physical Habitat Simulation (PHABSIM) system of the IFIM generates habitat–discharge relations that are broad and flat (without sharp inflection points) for warmwater stream ecosytems. Such habitat–discharge relations are difficult to use in an assessment or management context. A similar analysis performed on a coldwater stream ecosystem generally produces more definitive results.

In summary, the basic assumption that stream flow is an important factor determining habitat quality for warmwater fishes is probably valid, although the precise manner in which river flow is related to habitat (or biomass) has not been described to the level of detail required for assessment of impact. Consequently, application of the IFIM to assess effects of river regulation or water withdrawal on warmwater fishes cannot be performed at a level of defensibility equivalent to that of coldwater stream applications until the unique hydrology and biology of these systems are described in greater detail and these findings incorporated into the IFIM.

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Determination of Instream Flow Needs at Hydroelectric Projects in the Northeast

by

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The Fish and Wildlife Service (Service) developed its New England Flow Method in 1981 in anticipation of reviewing numerous applications for new hydroelectric projects. At the time, there were about 10,000 existing dams in New England that were not being used for hydropower, but whose development was possible with the passage of the Public Utility Regulatory Policies Act of 1978. The New England Flow Method continues to be used by the Service, although its incorporation by the Federal Energy Regulatory Commission into hydroelectric project licenses and its effectiveness in protecting aquatic habitat have not been evaluated until now.

The New England Flow Method uses historical stream flow data to determine instream flow needs at a given hydroelectric project. The major biological assumption of the method is that instream flow equivalent to historical median discharge in August (base flow conditions) and during spawning and incubation periods will protect aquatic habitat to the degree that would be expected under a natural flow regime. Recommended flows are derived from historical flow data for the river on which the project is located. In certain circumstances, regional historical flow data are used, based on gaging information for unregulated rivers throughout New England. The regional median August flow is equivalent to 0.07 $m^3/s/km^2$ of drainage area (0.5 $ft^3/s/mi^2$).

Using the New England Flow Method, the median August flow (Aquatic Base Flow) is recommended as the instream flow requirement that applies throughout the year unless higher flows are needed on a seasonal basis for migration, spawning, or egg incubation. The method also allows for project releases to equal inflow to the project area, when the latter are lower than the prescribed flows. Inflows lower than historical monthly median discharges would be expected during natural drought conditions or when upstream projects are storing water for peaking or flow augmentation.

Since its inception in 1981, the New England Flow Method has been the basis for instream flow requirements at 157 hydroelectric projects in New England (77% of a total 205 projects authorized by the Federal Energy Regulatory Commission). At 48 projects (23%), the Commission used alternate means for determining instream flows, including requiring run-of-river operation, which obviates the need for minimum flows.

Lack of data on baseline and postproject conditions precludes direct observation of the level of habitat protection provided by flows derived from the New England Flow Method. However, an indirect measure can be found by examining the results of 14 instream flow studies. These studies were conducted by project proponents between 1981 and 1988 and used habitat-based techniques, including the Instream Flow Incremental Methodology. Flows supported by the results of the instream flow studies ranged from 63% below to 90% above those derived from the New England Flow Method (median 11.1% below the Aquatic Base Flow). In all but three cases, the instream flow studies gave results that were lower than corresponding values for Aquatic Base Flow. These results suggest that the New England Flow Method produces conservative results, favoring aquatic resource protection.

Review of numerous hydroelectric projects under prescribed deadlines requires an efficient approach. Faced with a high workload and limited resources, Service biologists often need to quickly develop recommendations for instream flows that will adequately protect fish and wildlife resources. The New England Flow Method can be applied with minimal information and no field studies. In addressing issues related to minimum flows at hydroelectric projects

in New England, the method appears to meet the Service's needs.

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Discharge versus Habitat in Steep-gradient Piedmont Streams

by

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The focus of previous, ongoing, and future stream fish habitat research is the relation of the fish community to substrate composition, water depth, and water velocity. My basic premise has been that a diverse habitat will support a diverse fish community if water quality is suitable. After many years we have come to expect reductions in fish species diversity following catastrophic changes to a stream's habitat, such as channelization. Fishes of the Piedmont in the southeastern United States may be divided into three broad categories of habitat users: sculpins (*Cottus* spp.) and darters (*Etheostoma* spp. and *Percina* spp.), which prefer swift waters of riffles; minnows and suckers, which use open runs and midstream pools; and sunfishes (*Lepomis* spp.) and basses (*Micropterus* spp.), which seek cover near undercut banks and jams. These three broad categories of habitat are defined largely in terms of water velocity and depth. Substrate diversity is also important, not only because it provides a variety of spawning habitat for fishes, but also because it is the primary site of aquatic insect productivity.

The ratio of riffles to pools is thought to represent a fundamental characteristic of a stream ecosystem. A cross section or transect across a Piedmont stream or river frequently produces a series or mixture of small pools and riffles. Whereas a pool by itself might serve as a feeding station for a predatory fish, suitability of the pool depends on its proximity to a riffle in order to provide drifting insects. The original riffle–pool concept was theorized as a longitudinal phenomenon; however, a balance of riffles and pools at an acceptable level (e.g., at a ratio equal to that at median flow) might very well be necessary for maintenance of a diverse fish community.

Longitudinal increases in fish species diversity are well known. In my research, fish species diversity was predictable from substrate diversity, and substrate diversity increased downstream. Fish abundance (i.e., catch per unit effort) was positively related to width, depth, and percent cobble (i.e., rubble) and negatively related to percent bedrock and percent silt and sand in the substrate. Currently, I am researching (1) the relation between discharge and diversity of velocity–depth combinations, (2) the relation between discharge and the riffle–pool ratio, (3) how relations 1 and 2 are related to conventional wetted perimeter–discharge curves, and (4) fish community diversity in relation to velocity–depth diversity and the riffle–pool ratio. Field methods involve placing numerous transects in a study section and measuring velocity, depth, and substrate at 1-m intervals along each transect. Subsequently, each interval is categorized into 1 of 13 categories, which include dry, 3 riffle categories, 5 pool categories, and 4 categories of runs. Velocity–depth diversity seems to gradually decline with reductions in discharge, and it collapses near the inflection point on wetted perimeter–discharge curves. This reduction is primarily due to reduction in the 4 categories of runs. The riffle–pool ratio remains generally flat with reductions in discharge, but collapses near the wetted perimeter–discharge inflection point.

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Use of Multiple Methods for Instream Flow Recommendations—A State Agency Approach

by

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North Carolina's waters range from the coldwater trout streams of the Appalachian Mountains to the meandering blackwater streams of the Coastal Plain. In between lie the coolwater streams and rivers of the Foothills and the Piedmont's broad, turbid rivers. The State's instream flow issues are also diverse. Many of these issues involve hydropower projects—either the existing facilities or the planning and design of new ones—and include peak power production, diversions of various lengths, and run-of-river projects that illegally peak. There are also concerns with existing reservoirs constructed for purposes such as municipal and industrial water supply, recreation, or irrigation. Many of these projects are old, and most do not have a required minimum flow. Direct withdrawals of water from streams also affect instream flow and aquatic habitat, including withdrawals for municipal and industrial water supply, irrigation, aquaculture, and thermal power plant cooling water.

The Division of Water Resources has been involved in instream flow studies since 1977. The division has conducted over 50 wetted perimeter studies and 24 Instream Flow Incremental Methodology (IFIM) studies, and it has made numerous flow recommendations using desktop methods. Every available opportunity is taken in project evaluation to address instream flow concerns. The State Dam Safety Law is applicable to all non-Federal dams higher than 15 feet that impound at least 10 acre-feet of water. This law is used to stipulate minimum flows from new dams unless they are located on intermittent streams. Hydropower projects must obtain a Certificate of Public Convenience and Necessity (CPCN) from the North Carolina Utilities Commission before construction. If the hydropower project will not be receiving a license or exemption from the Federal Energy Regulatory Commission, then the Division of Water Resources will attach its instream flow recommendation to the Utilities Commission certificate. Instream flow recommendations may also be added as conditions to State 401 water quality certification and Federal 404 permits for projects that affect stream flow regimes.

Many factors are considered in selecting an instream flow assessment method. First, the value of the resource to be protected is considered. For example, a prime redbreast sunfish stream is handled differently than a severely degraded stream. The magnitude and duration of the project's impact are also evaluated by such factors as distance to the next major tributary, length of the diversion, or number of acres of potentially affected habitat. An attempt is made to determine whether the project will become controversial and what type of negotiations might take place. Other concerns include the manpower available for a study and how much time remains before a decision must be made.

The instream flow methods used can be separated into desktop and field methods. Desktop methods are used for projects with small impact or when a recommendation must be made quickly. A reconnaissance visit is usually made, and a recommendation is developed using one of several desktop methods. A previous recommendation may be ratioed by drainage area to a new location if there is an existing instream flow study site nearby and if the initial field visit indicates that the projects are similar. Review of hydrologic data may be used to develop a recommendation. Stream flow data from an appropriate U.S. Geological Survey gage are analyzed for mean annual flow and 7Q10 (lowest flow for 7 consecutive days, with a 10-year recurrence interval), as well as monthly means, medians, and lowest daily flows of record. These data are used to compare natural and proposed flow regimes. In areas where aquatic habitat is deemed minimal or nonexistent, the 7Q10 flow becomes the instream

flow recommendation. No flow recommendations have been less than the 7Q10 flow.

The New England Method, as adapted for use in North Carolina, uses the September median daily flow as the instream flow recommendation. The standard recommendation from the North Carolina Wildlife Resources Commission and the U.S. Fish and Wildlife Service is that a flow equal to the September median be provided, or a site-specific instream flow study must be conducted.

Field methods are used to develop instream flow recommendations when potential impacts are high, the stream contains a high quality fishery, or the project will involve peak power production. A site-specific study may be conducted by a developer to determine if an earlier desktop recommendation can be lowered or to provide answers to "what if" questions during negotiations. Field study methods used by the Division of Water Resources include wetted perimeter, regression model, incremental wetted surface area, and the IFIM. A State and Federal interagency team conducts a site visit and determines the most appropriate field methods to be used.

Wetted perimeter is one of the more frequently used field methods. Stream cross-sections, or transects, are selected to represent all habitat types contained in the stream reach of interest. The bottom profile of each transect is surveyed relative to a benchmark. Stream discharge and water surface elevations are measured at a minimum of three different flows so that a stage–discharge relation can be developed for each transect. Plots of transect wetted perimeter and stage are used to find the point of inflection (water surface elevation at which further reductions in stage result in large losses of wetted perimeter). The discharge corresponding to the water surface elevation of the point of inflection becomes the recommended flow for each transect. Through habitat mapping of the stream reach of interest, the percentage of the stream represented by each transect or the stream reach coefficient is determined. These stream reach coefficients and the sensitivity of each transect to stage reductions are used to weight each individual transect recommendation and develop the overall flow recommendation for the study site.

The wetted perimeter method is fairly inexpensive, and the time required for data collection can range from 2 days for a regulated stream to more than a year for remote sites on unregulated waters. Data analysis requires only 2 to 3 days. Because this method does not consider depth or velocity of water, any portion of the channel under water is considered habitat. Disadvantages of the method include subjective identification of point of inflection and the flow recommmendation representing only a single flow of the entire year. There is no way to evaluate the relation of flow to habitat at flows above or below the recommendation.

The regression model method is another field approach that was developed to provide a wetted perimeter recommendation. Numerous parameters from over 50 study sites across the State were analyzed for their correlation with the wetted perimeter recommendation at each site. To develop an instream flow recommendation, the model requires only mean annual flow, 7Q10, and average width at wetted area–discharge point of inflection. Validation analyses for new sites showed that the model predicts well for Piedmont streams. This method requires only one field visit to select and survey transects. No discharge measurements are needed, and the survey data can be collected during any season except during high water. The disadvantages are the same as for the wetted perimeter method.

The incremental wetted surface area method was developed as an improvement to the wetted perimeter method. Field data collection is the same as for the wetted perimeter method. For a given discharge, this method calculates the wetted surface area at each transect (wetted perimeter × length of stream represented by the transect), and the procedure is repeated for all flows of interest. Plots of wetted surface area versus discharge are developed for each transect and the entire study site to indicate flow versus habitat relation. The point of inflection on a plot of wetted surface area for the overall study site is usually the recommended flow. Individual transect plots can be checked to determine the amount of wetted surface area at the recommended or other flows. The time needed to complete field data collection for this method is the same as for the wetted perimeter method; data analysis then requires 4 to 5 days. The main advantage of the incremental wetted surface area method is that it provides an indication of how wetted surface area changes over a wide range of flows. Its main disadvantage is that any immersed channel is considered habitat.

The IFIM is considered the state-of-the-art method for instream flow analyses and is widely accepted and used. It is used for projects that (1) are expected to have significant impacts, (2) may affect an outstanding fishery, or (3) are proposed as peaking hydropower producers. It is also used where complicated negotiations will be required to arrive at the recommended stream flow regime. Depth, velocity, substrate, and cover are all used in developing the flow versus habitat relation. The IFIM requires more field time and considerably more data analysis time than previously described field methods. Conse-

quently, staff resources from agencies will be high, even if much of the work is conducted by a private consulting firm.

The availability of several methodologies for evaluating and recommending instream flows provides a multitiered approach to solving instream flow problems. Each higher method offers a more refined recommendation but requires more investment of time and resources by the agency. The level at which the return no longer justifies the effort will differ for each instream flow situation. Agencies in North Carolina try to select the approach that most effectively addresses a particular case. If the situation changes, there is nothing to preclude advancing to a more complex technique.

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Stream Habitat Analysis and Instream Flow Assessment: A State–Federal Effort in Arkansas

by

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Recent legislation passed by the Arkansas Legislature requires the determination of instream flow requirements for beneficial uses in the State's major rivers. In response to this mandate, the Arkansas Game and Fish Commission and State Department of Pollution Control and Ecology developed the Arkansas Method of instream flow determination for fish and wildlife needs. The Arkansas Method is a modification of the Tennant Method (Tennant 1975, 1976) for instream flow assessment; it uses flow standards for three seasonal periods. Sixty percent of the mean monthly flow (MMF) for November through March is recommended for flushing sediments and shaping channels. Seventy percent of the MMF for April, May, and June is recommended to protect spawning. Fifty percent of the MMF (or median, in some cases) for July to October is recommended to maintain water quality and provide habitat conditions conducive to fish growth and production. The flow standards were developed from information on historic streamflows in Arkansas, field experience and data on stream fishes, and knowledge of natural seasonal processes.

Because many of the State's highest quality streams are found on National forests, a cooperative State–Federal study was developed to assess stream flows in relation to fish population abundance, habitat, and water chemistry. Field studies were conducted from 1985 to 1988 in third- through fifth-order streams in the Ozark–St. Francis and Ouachita National Forests. Stream reaches were delineated by channel morphology and substrate, and entire pools or riffles were sampled at least three times each year, corresponding to Arkansas seasonal flows.

The structure of the fish community in each habitat type was summarized using feeding guilds, percent composition per species, and biomass per family group. Stream habitat was surveyed at each site using 24 habitat measurements, and water quality and instream flow measurements were made. The number of species generally increased from headwaters to midreaches and was in most cases associated with addition of new species rather than replacement. The majority of species added with increasing stream order were pool species or slow-water, large-channel species. Habitat diversity increased with downstream progression, canopy closure decreased as channels became wider, substrate particle size became more heterogenous, and

mean riffle–pool depth increased. Pool–riffle ratio in most Ozark and Ouachita streams remained constant (1:1) with progression to midreaches.

The studies were initiated to determine if flow recommendations computed using the Arkansas Method were appropriate for smaller-order streams of the Ozark–St. Francis and Ouachita National Forests. The Arkansas Method was found to be adequate to protect the stream fisheries in National forest watersheds. However, for streams that regularly have only subsurface flow during dry periods (i.e., intermittent as measured by streamflow gages), the Arkansas Method may need to be refined.

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Bain, Mark A., editor. 1990. **Ecology and Assessment of Warmwater Streams: Workshop Synopsis.** U.S. Fish Wildl. Serv., *Biol. Rep.* 90(5). 44 pp.

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Key words: Stream ecology, impact assessment, habitat requirements, instream flow, macroinvertebrates, fish communities.

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